

Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015



Megan P. Singleton^{a,*}, Andrea E. Thode^a, Andrew J. Sánchez Meador^a, Jose M. Iniguez^b

^a Northern Arizona University, School of Forestry, PO Box 15018, Flagstaff, AZ 86011, United States

^b USDA Forest Service: Rocky Mountain Research Station, 240 West Prospect Rd, Fort Collins, CO 80526, United States

ARTICLE INFO

Keywords:

Fire effects
MTBS
RdNBR
Remote sensing
Fire severity
Arizona
New Mexico

ABSTRACT

In the last three decades, over 4.1 million hectares have burned in Arizona and New Mexico and the largest fires in documented history have occurred in the past two decades. Changes in burn severity over time, however, have not been well documented in forest and woodland ecosystems in the southwestern US. Using remotely sensed burn severity data from 1621 fires (> 404 ha), we assessed trends from 1984 to 2015 in Arizona and New Mexico in (1) number of fires and total area burned in all vegetation types; (2) area burned, area of high-severity, and percent of high-severity fire in all forest and woodland areas; and (3) area burned, area of high-severity, and percent of high-severity in seven different grouped forest and woodland vegetation types (Ecological Response Unit [ERU] Fire Regime Types). Number of fires and area burned increased across the Southwest regardless of vegetation type. The significant increasing trends held for area burned, area of high-severity, and percent of high-severity fire in all forest and woodland ecosystems. Area burned and area burned severely increased in all seven ERU Fire Regime Types while percent of high-severity fire increased in two ERUs: Mixed Conifer Frequent Fire and Mixed Conifer with Aspen/Spruce Fir. Managers must face the implications of increasing, uncharacteristic high-severity fire in many ecosystems as climate change and human pressures continue to affect fire regimes.

1. Introduction

Fire is an important and dynamic disturbance process, yet concerns over increasing extent and severity are reaching new urgency. Recent work across the globe has quantified trends in high-severity fire using remotely sensed technology in attempt to determine if historical fire patterns are changing (Archibald et al., 2010; Dennison et al., 2014; Kasischke and Turetsky, 2006; Picotte et al., 2016; Riaño et al., 2007; Rivera-Huerta et al., 2016). Shifts in fire regimes, especially in high-severity fire, can result in significant consequences to landscape processes and ecosystem function and the identification of such shifts are important for forest conservation and sustainability. In the western United States, land-use practices i.e., fire suppression and logging as well as climate change are frequently cited as the primary drivers of increasing fire severity (Dillon et al., 2011; Fulé et al., 2009, 1997; Jolly et al., 2015; Moore et al., 2004; Reilly et al., 2017; Williams et al., 2013). The southwestern US, namely Arizona and New Mexico, is a semi-arid region where forest structure has dramatically changed since Euro-American settlement (Moore et al., 2004; White and Vankjät,

1993) and where increased wildfire activity is known to be driven by climate change (Crimmins, 2011; Grissino-Mayer and Swetnam, 2000). The southwestern US has been recently affected by large and intense wildfires where 4.1 million hectares have burned in all vegetation types in the past three decades and the largest fires in documented history have occurred in the past two decades. Fires regimes may be shifting in the Southwest yet this phenomena has been largely undocumented. Severity is fundamental to understanding how fire patterns are changing and ultimately, in understanding the ecological implications of altered fire regimes.

Here we present a comprehensive region-wide trend analysis of high-severity fire in Arizona and New Mexico forest and woodland ecosystems over a 32 year time span using data from the Monitoring Trends in Burn Severity (MTBS) project. The MTBS project uses Landsat Thematic Mapper (TM), Enhanced Thematic Mapper Plus (ETM +), and Operational Land Imager (OLI) imagery to produce burn severity data for fires greater than 404 ha in the western US and 202 ha in the eastern US from 1984 to the present (Eidenshink et al., 2007). Dillon et al. (2011) examined temporal trends from 1984 to 2006 in high-

* Corresponding author.

E-mail addresses: mp754@nau.edu (M.P. Singleton), Andi.Thode@nau.edu (A.E. Thode), Andrew.SanchezMeador@nau.edu (A.J. Sánchez Meador), jiniguez@fs.fed.us (J.M. Iniguez).

<https://doi.org/10.1016/j.foreco.2018.11.039>

Received 9 September 2018; Received in revised form 19 November 2018; Accepted 20 November 2018

Available online 06 December 2018

0378-1127/ © 2018 Elsevier B.V. All rights reserved.

severity fire for parts of the Southwest, but excluded large areas of New Mexico and fires that occurred in unique areas of Southern Arizona (e.g., the Sky Islands). Dillon et al. (2011) also did not include MTBS products with post-fire images acquired less than 6 months from the fire ignition date (i.e., initial assessments [IA]) and thus, an estimated 43% of burned area was excluded from their analysis in the time frame of their study. There is a need to broaden the work of Dillon et al. (2011) by updating the time period, expanding the study area to encompass all of Arizona and New Mexico, and including all fires > 404 ha that burned since 1984 in order to provide a more complete and accurate burn severity dataset for the Southwest. There is also a need to evaluate temporal trends in high-severity fire within specific forest and woodland vegetation types to understand how recent wildfires have impacted individual ecosystems and to understand how fire patterns may be changing. Such information is critical in developing management strategies to increase the sustainability for each forest type.

Our objectives were to assess, from 1984 to 2015 in Arizona and New Mexico, if there were increasing trends in (1) number of fires and total area burned in all vegetation types; (2) area burned, area of high-severity fire, and percent of high-severity fire in all forested and woodland vegetation types (Ecological Response Units [ERU] Fire Regime Types); and (3) area burned, area of high-severity fire, and percent of high-severity fire in seven forest and woodland ERU Fire Regime Types.

2. Methods

2.1. Study area

Our study area encompassed all fires from 1984 to 2015 greater than 404 ha that burned within Arizona and New Mexico (Fig. 1). We included all fires that burned on private, state, and federal lands including Forest Service (FS), National Park Service (NPS), Bureau of Land Management (BLM), Fish and Wildlife Service (FWS), Department

Table 1

Area burned, total forested land, and percent of burned forested land from 1984 to 2015 by management agencies in the study area. The 'other' category includes forested lands not designated as federal land i.e. state and private lands.

Ownership	Forested area burned (ha)	Total forested land (ha)	Percent burned (%)
DOD	7570	90,658	8.4
BIA	297,662	3,124,011	9.5
BLM	70,760	1,191,671	5.9
FS	1,483,239	6,669,639	22.2
NPS	66,076	259,325	25.5
FWS	651	17,601	3.7
OTHER	209,094	5,556,263	3.8

of Defense (DOD), and Bureau of Indian Affairs (BIA) managed lands (Table 1). We assessed 1621 fires that burned in all vegetation types and 1143 fires that burned in forests and woodlands. Elevations in the study area ranged from 66 m to 3647 m as recorded by the lowest and highest pixel in the dataset from digital elevation models. The study area is characterized by a semi-arid climate but climate over much of the Southwest is highly variable due to its broad range of topographic features and its location near the Gulf of Mexico, the Gulf of California, and the Pacific Ocean (Sheppard et al., 2002). Mean annual winter temperatures range from 0 °C to 14.3 °C and mean annual summer temperatures range from 17.7 °C to 33.5 °C (at the lowest and highest elevations in the Southwest) from 1984 to 2015 (NOAA, 2018). The Southwest is also characterized by monsoonal rains occurring from July–September and receives up to 50% of its rainfall during this time (Sheppard et al., 2002). From 1984 to 2015, Arizona received an annual average rainfall of 30.6 cm while New Mexico received an annual average rainfall of 36.8 cm (NOAA, 2018). Another important source of moisture is late-spring/early-summer snowmelt which is critical for plant growth (Notaro et al., 2010). Thus, dual peaks of moisture occur throughout the year resulting in a bimodal seasonal cycle of vegetation

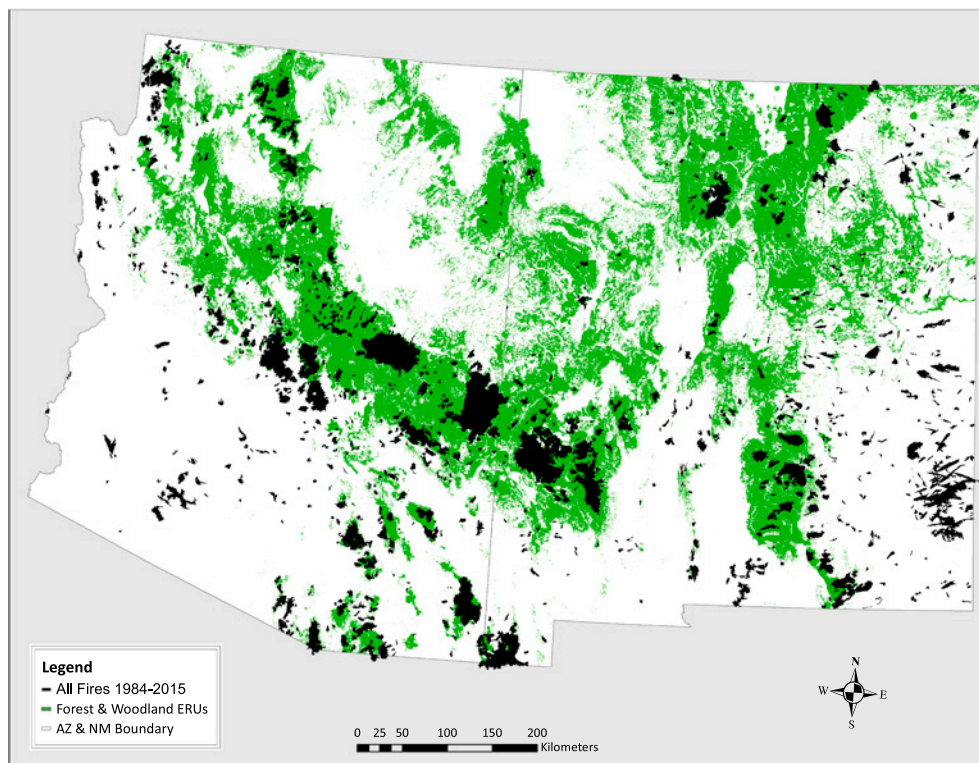


Fig. 1. Map of the study region, fires analyzed in this study, and all forest and woodland Ecological Response Units. Black polygons represent 1621 fires that were analyzed in the study area.

Table 2

Table showing the 12 forest and woodland Ecological Response Units (ERU) grouped into seven ERU Fire Regime Types (Barrett et al., 2010; Wahlberg et al., 2014).

ERU	ERU Fire Regime Types
Madrean Encinal Woodland	Madrean Encinal Woodland & Madrean
Madrean Pinyon-Oak Woodland	Pinyon-Oak Woodland
PJ Grass	PJ Grass & Juniper Grass
Juniper Grass	
PJ Evergreen Shrub	PJ Evergreen Shrub
PJ Sagebrush	PJ Sagebrush & PJ Woodland
PJ Woodland	
Ponderosa Pine Forest	Ponderosa Pine
Ponderosa Pine - Evergreen Oak	
Mixed Conifer - Frequent Fire	Mixed Conifer - Frequent Fire
Mixed Conifer w/Aspen	Mixed Conifer w/Aspen & Spruce-Fir Forests
Spruce-Fir Forest	

greenness after spring-summer snow melt and summer/early-autumn monsoon rainfall (Notaro et al., 2010; Sheppard et al., 2002).

2.2. Vegetation layer

We identified forested and woodland areas using Wahlberg et al. (2014) Ecological Response Units (ERU) vegetation framework. This framework is the most current landscape stratification of vegetation classes for the Southwestern United States. Similar to LANDFIRE's Biophysical Settings (BpS) vegetation stratification (Rollins and Frame, 2006), ERUs are homogenous vegetation classes that share similar site potential and ecosystem processes and respond the same way under historic fire regimes (Wahlberg et al., 2014). Ecological Response Units are developed for landscape analysis and strategic planning purposes and built from the National Vegetation Classification (NVC) and the Terrestrial Ecological Unit Inventory (TEUI) frameworks.

We used a total 12 of the 13 ERUs that were classified as forest or woodland, excluding Bristlecone pine ERU from our analysis due to the low area burned in the time frame of the study. We aggregated similar ERUs that shared similar historic fire regimes (Wahlberg et al., 2014) that we call ERU Fire Regime Types (Table 2/Fig. S1). To define each ERU's fire regime, Wahlberg et al. (2014) uses information on fire return intervals from published literature as well as LANDFIRE fire regime classifications from the Fire Regime Condition Class Handbook (Barrett et al., 2010).

Similar to other studies conducting trend analyses (Dillon et al., 2011; Miller and Safford, 2012), we used a vegetation map that incorporates natural disturbance fire regimes prior to European settlement instead of using existing vegetation maps. Existing vegetation maps are likely to exclude greater proportions of forested and woodland areas in earlier years of the study because high-severity fire and multiple entries can cause changes in vegetation types over time. Existing vegetation maps may also underestimate the occurrence of high-severity fire in earlier years of the time series and introduce a phantom trend (Hanson and Odion, 2014). Therefore, we used a potential vegetation map that classifies vegetation based on similar ecosystem characteristics and when natural disturbances prevail. The minimum mapping unit for the ERU data is five hectares. Although our burn severity data resolution is 30 × 30 m, we did not use ERU data for spatial analysis. Rather, we stratified ERU by Fire Regime Type and used it as reference boundaries in determining the vegetation type for burned pixels.

2.3. Burn severity mapping

To investigate recent trends we used all data from the Monitoring

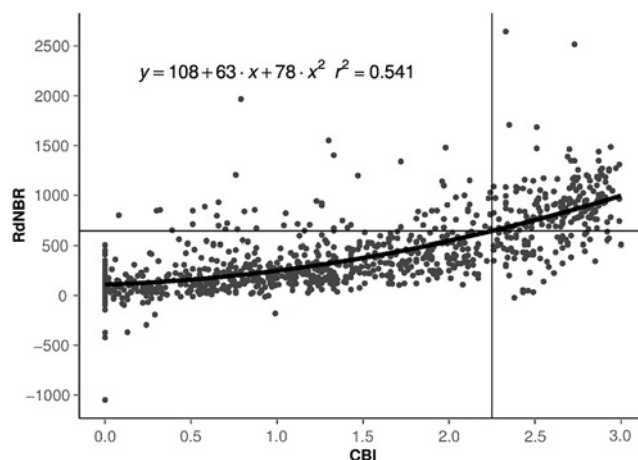


Fig. 2. Quadratic regression model of the relative differenced normalized burn ratio (RdNBR) versus 1062 field-measured composite burn index (CBI) plots. Crossed lines show the threshold for high severity (CBI = 2.25; RdNBR = 643).

Trends in Burn Severity (MTBS) program from 1984 to 2015 in Arizona and New Mexico (MTBS, 2013). We used the relative differenced normalized burn ratio (RdNBR) images to account for change relative to the amount of pre-fire vegetative cover and to make comparisons of fires across space and time (Miller and Thode, 2007). To determine a high-severity RdNBR threshold for the Southwest, we used the value of 643 obtained from a regression model of 1197 Southwest Composite Burn Index (CBI) field plot data against their corresponding RdNBR values (Fig. 2) (Gdula and Brannfors, 2014). The CBI plots were conducted in the Grand Canyon National Park, Kaibab, Coconino, and Gila National Forests and conducted in 8 out of the 12 forest and woodland ERUs used in this analysis. The CBI averages burn condition in 30 m diameter plot across five strata ((1) substrates, (2) herbs, shrubs and trees < 1 m, (3) tall shrubs and trees 1 to 5 m, (4) subcanopy poles/trees, and (5) dominant overstory trees) on a scale of 0 to 3 with lower numbers representing lower degrees of change and higher numbers represent higher degrees of change (Key and Benson, 2006). We calculated the RdNBR threshold equivalent to a CBI ≥ 2.25, representing a midpoint between high and moderate severity classes (Miller and Thode, 2007).

To produce a complete burn severity dataset for the Southwest, we reprocessed 718 MTBS initial assessment [IA] products to one year post-fire extended assessments [EA] and reanalyzed 85 extended assessments with poor pre- and/or post-fire scene selections. To do this we utilized the MTBS Project's QGIS plugin, the Event Mapper Tool, which allowed us to remap and include every recorded fire in our analyses (MTBS, 2013). We selected post-fire scenes that were acquired 6–24 months following the fire (Dillon et al., 2011) to ensure that the post-fire image did not over-represent severity (i.e., < 6 months) and then to ensure that severity was not underrepresented (i.e., > 24 months). We selected pre-fire images that were ≤ 2 years to ensure that no other disturbance events interfered with the burn signal. We define severity as the effect of fire on dominant vegetation 6–24 months post-fire relative to ≤ 2 year pre-fire conditions (Dillon et al., 2011). To account for phenological differences in the pre- post-fire images, we excluded fires with an offset value ≥ 100 (Stephen Howard, personal communication). The offset is the average dNBR value of pixels outside of a homogenous area of the burn perimeter in a post-fire image differenced by the same area in a pre-fire image that is applied to RdNBR images (Key, 2006). If pre- and post-fire images are perfectly matched the dNBR offset value is zero (Cansler and McKenzie, 2014; Key, 2006). We also applied a 3x3 focal mean to all RdNBR images to minimize pixilation (Miller et al., 2012). We combined our 803 remapped fires with 340 of MTBS Projects' extended assessment products to analyze temporal trends in high-severity fire.

2.4. Trend analysis

We examined trends in number of fires and total area burned for all vegetation types over the 32 year period. We also examined trends in total area burned, high-severity area, and percent of high-severity fire for fires that burned in forests and woodlands and for each of the seven ERU Fire Regime Types (Table 2/Fig. S1). We analyzed the number of fires by summing the count of all fires > 404 ha by year. We assessed area burned by summing the total pixels burned per year and area burned severely by summing the total of high-severity pixels ($RdNBR \geq 643$) per year. We calculated percent of high-severity fire by dividing the summed count of all high-severity pixels by the summed count of all burned forested pixels for each year.

Our data contained variables measured sequentially in time at fixed intervals, known as time series data (Metcalf and Cowpertwait, 2009). Similar to other studies, we used ARMA (Autoregressive Moving Average) time series regression methods to account for temporal autocorrelation and to determine if there were any trends in fire count, area, and proportion data over time (Miller et al., 2012, 2009; Miller and Safford, 2012; Stephens, 2005). An ARMA (p,q) model (Eq. S1) has an autoregressive component (up to order p) that refers to the use of past values in the regression equation, and a moving average component (up to order q) that represents the error of the model as a combination of previous error terms (Stephens, 2005). To identify the best-fit model, we used the Box-Jenkins technique, which involves identifying ARMA time series models, estimating its parameters, and diagnosing the model (Box and Jenkins, 1976).

To stabilize heteroscedasticity, we natural log-transformed fire count data and area data, and arcsine-square root transformed all percent data before model identification (Sokal and Rohlf, 1981). To identify appropriate models, we took the first difference of the data (Eq. S2) to remove any linear trends and to form a stationary series (Metcalf and Cowpertwait, 2009). Stationary data is important for model identification because the mean, variance, and autocorrelation structure do not change over time.

We identified patterns in the differenced data's autocorrelation functions (ACFs) and partial autocorrelation functions (PACFs). If an autocorrelation or partial autocorrelation at any given lag is significantly different than zero, the correlation is included in the ARMA model (Metcalf and Cowpertwait, 2009). When ACFs and PACFs indicated no significant autocorrelations, we used linear models to analyze the trend. To compare competing ARMA models, we used the corrected Akaike Information Criterion (AICc) where the model with the lowest AICc was chosen and when candidate models had a delta AIC ($\Delta AICc$) between 0 and 2, the more parsimonious model was selected (Shumway and Stoffer, 2010). We also used lowest root mean squared error (RMSE) in diagnosing model goodness-of-fit.

The time series data exhibited random walk with drift, where the drift component is in essence, the trend component. To determine significance of the trend, we calculated P-values of the drift coefficient and then back-transformed it into its native units. To calculate average increases per year for each of our datasets, we back-transformed the coefficient of the first and thirty-second observation from the fitted model and calculated the slope of the line. We also calculated mean, median, minimum and maximum for all variables.

3. Results

3.1. Trends in all fires from 1984 to 2015

In fires that occurred in all vegetation types, both fire count and area burned showed a significant increase from 1984 to 2015 ($P < 0.011$, $P \leq 0.001$) (Table 3, Figs. 3A/B, S2A/B). The average annual increase for area burned was 8373 ha (Table S1).

In forests and woodlands, ARMA and linear models for area burned, area burned severely, and percent burned severely showed significant

increasing trends ($P < 0.044$) (Table 3). The average annual increase of area burned and area of high-severity fire was 6360 ha/year and 1009 ha/year, respectively while the annual average increase in percent high-severity fire was 0.33% (Table S1). Trends in area burned in all vegetation types and trends in area burned and area burned severely in forests and woodlands showed a distinct increase after 2000 (Figs. 3B/C/D, S2B/C/D). Trends in percent high-severity fire were more variable yet consistently increased throughout the 32 year time period with percent peaking in years 1990, 2000, 2002, 2004 (Figs. 3E, S2E).

3.2. Trends in ERU fire regime types from 1984 to 2015

Over the 32 year time span, area burned ranged from approximately 146,000 to 442,000 ha and area burned severely ranged from 20,000 to 116,000 ha across all ERU Fire Regime Types (Table 4). Mixed Conifer with Aspen/Spruce Fir had the highest percent of its landbase burned severely (28%) while Madrean and PJ Grass/Juniper Grass had the lowest (8%) (Table 4). All ARMA and linear models for all ERU Fire Regime Types for area burned and area of high-severity fire showed a significant increasing trend ($P \leq 0.005$, $P \leq 0.016$) (Table 3). Percent of high-severity in Mixed Conifer Frequent Fire and Mixed Conifer w/ Aspen and Spruce Fir also showed a significant increasing trend ($P < 0.001$). We did not find significant temporal trends in percent of high-severity fire in Madrean, PJ Grass/Juniper Grass, PJ Evergreen Shrub, PJ Sagebrush and PJ Woodland, and Ponderosa Pine ($P \geq 0.126$). Increases in annual area burned and area burned severely in all ERU Fire Regime Types ranged from 400 to 2311 ha and 14–247 ha respectively while annual increases in percent of high-severity fire ranged from 0.08 to 0.84% (Table S1). Although increases in percent are seemingly low, they are also compounding. For example, an increase of 1% a year in Mixed Conifer with Aspen and Spruce Fir ERU Fire Regime Type equates to 32% over the 32 year time period.

Results in area burned and area burned severely also show a marked increase after the year 2000 (Figs. 4, 5, S3 and S4). Trends in area burned and area burned severely in PJ Sagebrush and PJ Woodland begin earlier i.e., 1996 (Figs. 4D, 5D, S2D and S4D). Similar to results for percent of high-severity fire in all fires in forest and woodlands, trends in percent of high-severity fire by ERU Fire Regime Type are variable with no discernible years where trends begin to increase (Figs. 6 and S4).

4. Discussion

Over the last three decades, fires are more frequent, larger, and more severe in the Southwest. The number of large fires (> 404 ha) and area burned increased from 1984 to 2015 across the Southwest regardless of vegetation type. The significant increasing trends held for area burned, area of high-severity fire, and percent of high-severity fire in forested and woodland ecosystems. In ERU Fire Regime Types, we found significant increases in area burned and area of high-severity fire across all seven types. Percent of high-severity fire showed significant increases in two out of the seven ERU Fire Regime Types, where Madrean, PJ Sagebrush/PJ woodland, PJ Evergreen Shrub, and Ponderosa Pine did not show an increase.

Our findings of increasing trends in number of fires, area burned, and in high-severity fire are similar to other studies using similar methodologies in different regions in the west (Dennison et al., 2014; Dillon et al., 2011; Miller et al., 2009; Miller and Safford, 2012). Dennison et al. (2014) conducted a broad-scale study across the western US examining temporal trends of large wildfires in nine ecoregions of similar climate variability and vegetation types from 1984 to 2011. They found that trends in fire size and area burned were most significant in the Arizona and New Mexico Mountain ecoregion. Miller et al. (2009) found increasing trends in percent of high-severity fire in all forest types from 1984 to 2006 in Sierra Nevada, CA. Similar to our findings, Miller et al. (2009) found an increase in percent of high-

Table 3

Table shows regression statistics for ARMA time series models and linear models, organized by All Fires 1984–2015 (top block) and then stratified by ERU Fire Regime Types from 1984 to 2015 (bottom block). Table shows ARMA and linear modeling statistics for number of fires, area burned in all vegetation types, area burned in forested and woodland ecosystems, area of high severity fire in forests and woodlands, and percent of high severity fire in forests and woodlands. All trends are significantly increasing in all vegetation types and in forests and woodlands. Table also shows area burned, area of high severity, and percent of high severity by ERU Fire Regime Type. Bolded *P* values are significant ($\alpha = 0.05$). All ERU Fire Regime Types show an increase in area burned and area of high severity. Mixed conifer Frequent Fire and Mixed Conifer with Aspen/Spruce-Fir showed an increase for percent of high severity fire.

			Best Model	AICc	RMSE	adj R^2	drift	<i>P</i> value
All Fires 1984–2015	All Veg Types	Frequency of Fires	Linear			0.017		0.011
		Area Burned	(0,1)	86.58	0.806		0.076	<0.001
	Forests and Woodlands	Area Burned	(0,1)	91.06	0.876		0.107	<0.001
		Area of High Severity	(1,0)	145.3	2.017		0.178	0.002
		Percent of High Severity	Linear			0.099		0.044
ERU Fire Regime Types	Madrean	Area Burned	Linear			0.247		0.002
		Area of High Severity	Linear			0.153		0.016
		Percent of High Severity	(0,1)	-29.93	0.130		0.002	0.521
	PJ Grass & Juniper Grass	Area Burned	(0,1)	96.25	0.939		0.085	<0.001
		Area of High Severity	(0,1)	143.96	1.975		0.165	0.002
		Percent of High Severity	(1,0)	-30.58	0.129		0.005	0.310
	PJ Evergreen Shrub	Area Burned	Linear			0.402		<0.001
		Area of High Severity	(1,1)	147.56	1.999		0.146	0.013
		Percent of High Severity	(0,1)	-9.48	0.180		0.004	0.319
	PJ Sagebrush & PJ Woodland	Area Burned	(0,1)	129.76	1.584		0.164	<0.001
		Area of High Severity	(1,1)	143.9	1.886		0.175	0.004
		Percent of High Severity	(1,0)	-2.77	0.200		0.007	0.161
	Ponderosa Pine	Area Burned	(0,1)	95.79	0.932		0.110	<0.001
		Area of High Severity	(1,1)	143.59	1.881		0.156	0.003
		Percent of High Severity	Linear			0.006		0.283
	Mixed Conifer Frequent Fire	Area Burned	(1,1)	132.69	1.582		0.166	0.005
		Area of High Severity	(0,1)	150.59	2.198		0.196	<0.001
		Percent of High Severity	(0,1)	-10.67	0.177		0.010	0.001
	Mixed Conifer w/Aspen & Spruce Fir	Area Burned	(1,1)	146.75	1.975		0.173	<0.001
		Area of High Severity	(1,1)	158.47	2.299		0.195	<0.001
		Percent of High Severity	(0,1)	6.53	0.231		0.004	0.001

severity fire in mixed conifer forests but did not observe an increase in ponderosa pine. When [Miller and Safford \(2012\)](#) expanded on their previous work and combined mixed-conifer and yellow pine into one grouping, they found increasing trends in area and percent of high-severity fire from 1984 to 2010. Although ponderosa pine and xeric mixed conifer share similar fire attributes in frequency and severity, we felt it more meaningful to examine each forest type separately in our analysis to provide information on fire effects in specific ecosystems that are of a concern to land managers. [Miller and Safford \(2012\)](#) did not find significant increasing trends in red fir, which contrasts to our findings in Southwestern higher elevation forests.

Our results are comparable to [Dillon et al. \(2011\)](#), especially since they assessed temporal trends in Southwestern regions. [Dillon et al. \(2011\)](#) found a significant increase in area burned and in area of high-severity in all three Southwest ecoregions (i.e., Southern Rockies, Colorado Plateau, Mogollon Rim) but only found an increase in proportion of high-severity in the Southern Rockies ecoregion. Similar to [Dillon et al. \(2011\)](#) we found a significant increase in area burned and area burned severely but one notable difference is that we found an increase in percent of high-severity fire across the whole region. We speculate that [Dillon et al. \(2011\)](#) did not find increasing trends in proportion of high-severity fire in the Southern Rockies and Colorado Plateau

ecoregions for a few reasons. First, the time span of their analysis was nine years shorter than ours and big fire years occurred in the Southwest after 2006. In our analysis, we recorded 16% of high-severity fire in 2007, 16% in 2012 and 18.5% in 2013, which could have driven trends upward. Further, they did not include initial assessment fires (products with post-fire images < 6 months from the ignition date) in their study, which excluded 375,644 ha of forested burned area and 28,344 ha of high-severity fire from their Southwest regional analysis from 1984 to 2006. Third, due to their pre- and post-fire image constraints, they excluded an additional 160,408 ha of forested burned area and 22,648 ha of high-severity. In comparison, we did not exclude any recorded fires in our dataset and have a more spatially and temporally comprehensive dataset than [Dillon et al. \(2011\)](#).

Similar to [Dillon et al. \(2011\)](#), we also observed a distinct shift post-2000 where area burned and area burned severely increase in our data. This shift may be a product of the relatively short time span of the study but could also be linked to climate and weather patterns. [Hoerling et al. \(2013\)](#) report that 2001–2010 was the warmest and fourth driest decade in the Southwest since 1901 with temperatures 0.8 °C higher than historic averages and a mean precipitation departure of −15 mm/year. Changes in climate have resulted in a severe drought that occurred in the first half of the 2001–2010 decade ([Hoerling et al., 2013](#)).

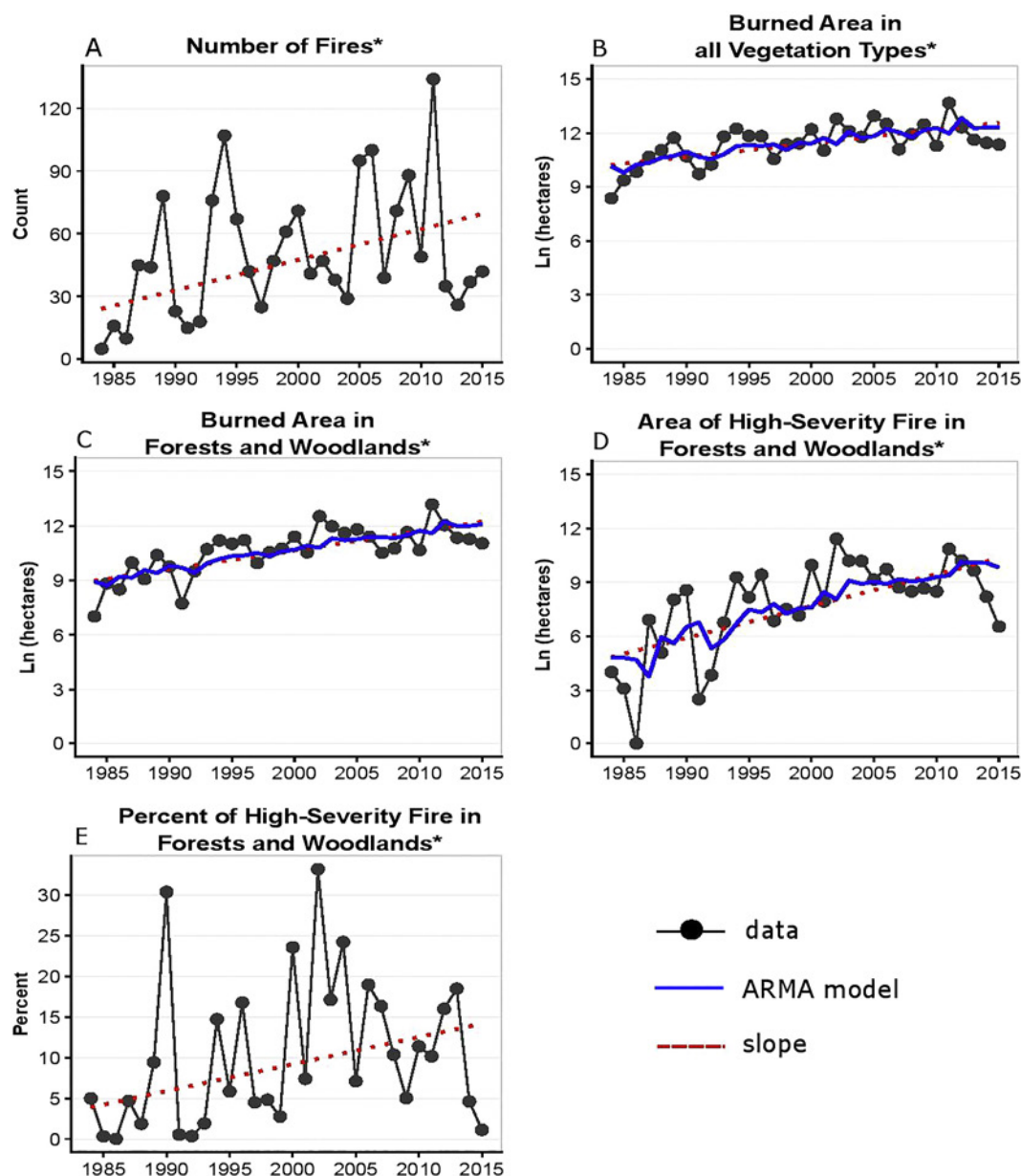


Fig. 3. Temporal trends in (A) number of large fires (> 404 ha), (B) burned area in all vegetation types, (C) burned area in forests and woodlands, (D) area of high severity in forests and woodlands, and (E) percent of high severity in forests and woodlands within Arizona and New Mexico from 1984 to 2015. *All trends are significant ($P \leq 0.044$).

Table 4

The 12 ERUs used in this study grouped by fire regime type showing area of high severity fire, area burned, and percent of high severity burned from 1984 to 2015. This table includes pixels that reburned from year to year.

ERU Fire Regime	Total area (ha)	Area burned (ha)	Area severe (ha)	Percent burned (%)	Percent high severity (%)
Madrean	1,002,125	318,465	25,239	32	8
PJ Grass/PJ Juniper Grass	6,266,846	393,180	32,667	6	8
PJ Evergreen Shrub	1,452,506	182,236	20,355	13	11
PJ Sagebrush/PJ Woodland	3,120,703	145,805	29,167	5	20
Ponderosa Pine	3,262,717	861,777	116,164	26	13
Mixed Conifer Frequent Fire	1,117,123	441,921	83,755	40	19
Mixed Conifer with Aspen/Spruce Fir	699,406	188,593	53,117	27	28

These drier conditions have increased the probability of severe fire and have been linked to increasing trends in fire size and area burned in the Arizona and New Mexico Mountain ecoregion (Dennison et al., 2014). The biggest fire years in recorded history within the Southwest occurred after 2000, where 2002 and 2011 resulted in the largest area

burned and largest area burned severely in forests and woodlands. With drought, earlier snowmelt, increases in summer and spring temperatures under a warming climate, fire years similar to 2002 and 2011 may not be the anomaly and trends in high-severity fire may continue to increase. Climate is known to play a role in driving the extent and

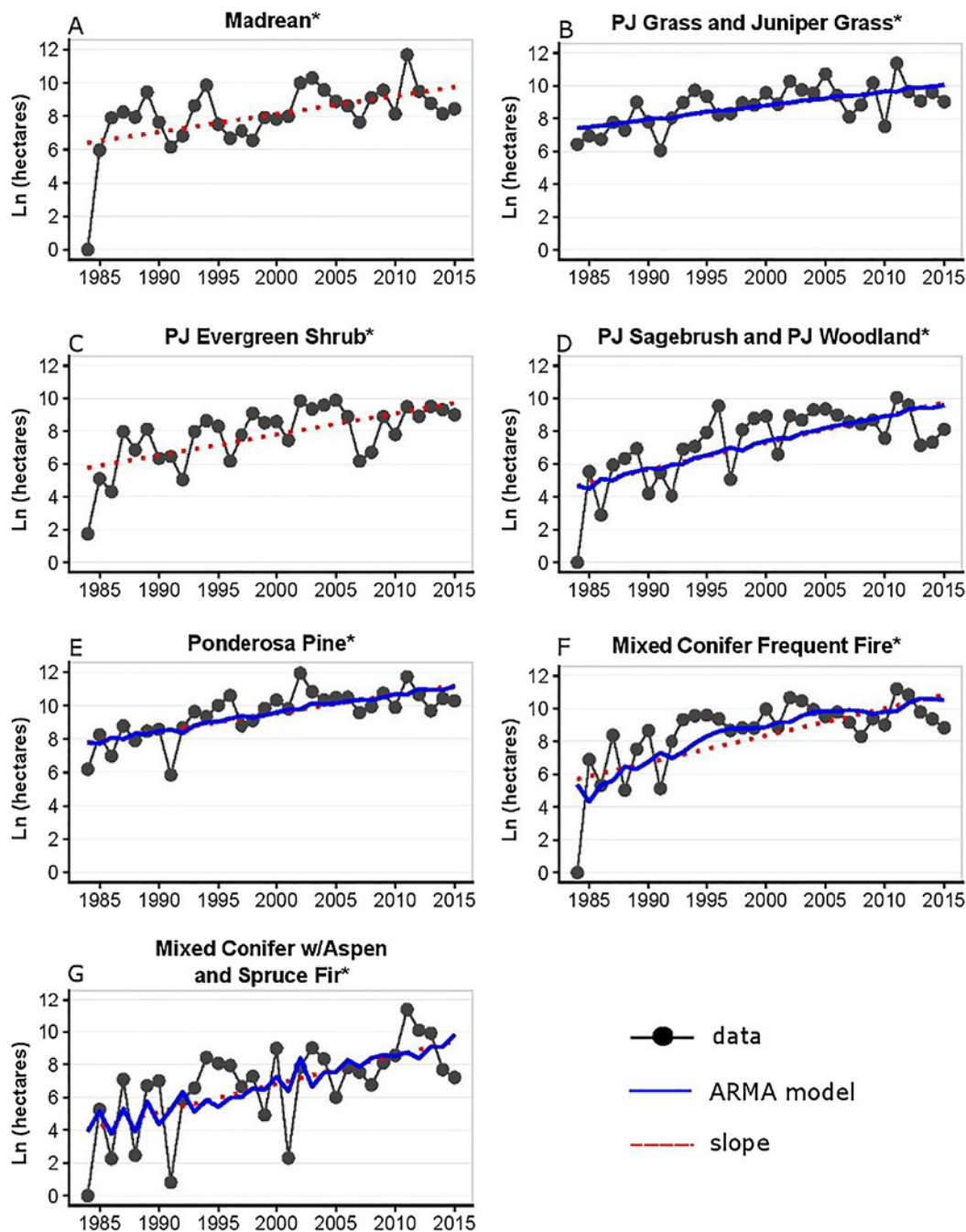


Fig. 4. Results in temporal trend analysis in area burned for each ERU Fire Regime Type from 1984 to 2015. *All trends are significantly increasing ($P \leq 0.005$).

frequency of fires (Westerling 2016, Williams et al., 2013) yet further work is needed to disentangle fire-climate relationships and in identifying specific climate drivers of high-severity in the Southwest.

Our study region was broad and encompassed a complex of different fire regimes within different land ownerships (Tables 1 and 4). Fire management practices vary greatly between federal, state and private land ownerships. Most notably, federal agencies (e.g., NPS and USFS) conduct prescribed burns and also allow lightning-ignited fire to burn to meet management objectives. Both of these management methods often yield less severe fire since certain weather prescriptions are set to limit the occurrence of extreme fire behavior. In our dataset we have included a total of 158 prescribed burns and 36 wildfire use burns (the management of naturally ignited fires to meet pre-defined resource objectives in a pre-defined area) with only one prescribed burn occurring prior to 1995 (MTBS, 2013). The number of wildland fire use fires

may be underreported in our dataset since the release of the 'Guidance for Implementation of Federal Wildland Fire Management Policy' in 2009. The revised policy allows naturally ignited wildland fires to be managed for multiple objectives instead of managing a fire to meet either suppression objectives or to provide natural resource benefits (Fire Executive Council, 2009). Therefore, a fire that has been suppressed in one area and allowed to burn in other areas may still be called a 'wildfire'. Nevertheless, other studies posit that prescribed and wildfire use fires skew high-severity trend results (Safford et al., 2015), yet we did not observe this in our study. Despite including fires that typically burn as low-intensity and low-severity surface fires in the last 18 years of the study period, we still observed significant upward trends.

Furthermore, we still found increasing trends despite not being able to adequately capture high-severity in 2011. On May 31, 2003 the Scan

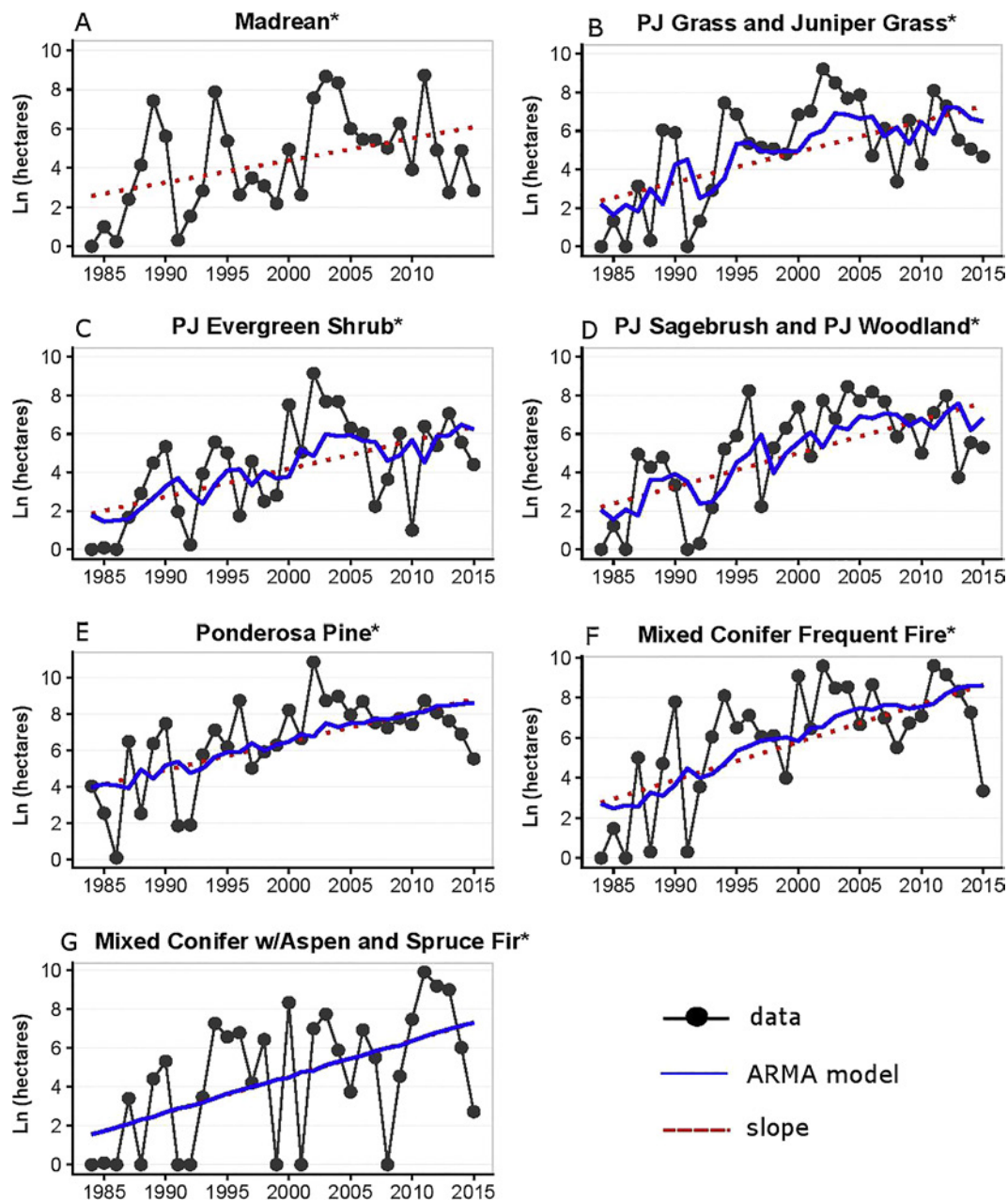


Fig. 5. Results in temporal trend analysis of area of high severity fire for each ERU Fire Regime Type from 1984 to 2015. *All trends are significantly increasing ($P \leq 0.016$).

Line Corrector (SCL) on Landsat 7 failed and subsequent imagery contained data gaps that resulted in a 22% loss of the scene (Chen et al., 2011). To circumvent this issue, we acquired post-fire scenes from Landsat 8 in 2013, two years after the fire date which significantly reduced the ability to properly capture high-severity in RdNBR images. Since 2011 was the Southwest's biggest recorded fire year, we would expect to see area of high-severity peaking and proportion of high-severity fire to be larger as well. Despite under-representing high-severity in a record-breaking year, these significantly increasing trends persist in the Southwest.

4.1. Management implications

This work provides quantitative evidence that total area burned and area of high-severity fire are increasing in Southwestern forest and woodland ecosystems across all types. Additionally, percent of high-

severity fire is increasing in two out of the seven ERU fire regime types.

Different ecosystems are adapted to, and characterized by varying fire size, spatial patterns, severities, and frequencies and that a disruption in any one of these fire regime attributes can result in large-scale changes in ecosystem function. It is likely that increasing trends in high-severity fire will threaten southwestern forest ecosystems. Many studies have already documented the consequence of uncharacteristic high-severity fire including an increase in high-severity patch size (Yocom-Kent et al., 2015), landscape homogeneity (Turner et al., 1994), altered plant species compositions (Turner et al., 1994), forest type-conversions (Savage and Mast, 2005), extinction of habitat (Lee et al., 2013), and altered soil properties and watershed dynamics (Neary et al., 1999).

Just as different ecosystems are characterized by different fire patterns, ecosystems will also differ in their response to increasing trends in severity. It is uncertain whether increasing trends in high-severity

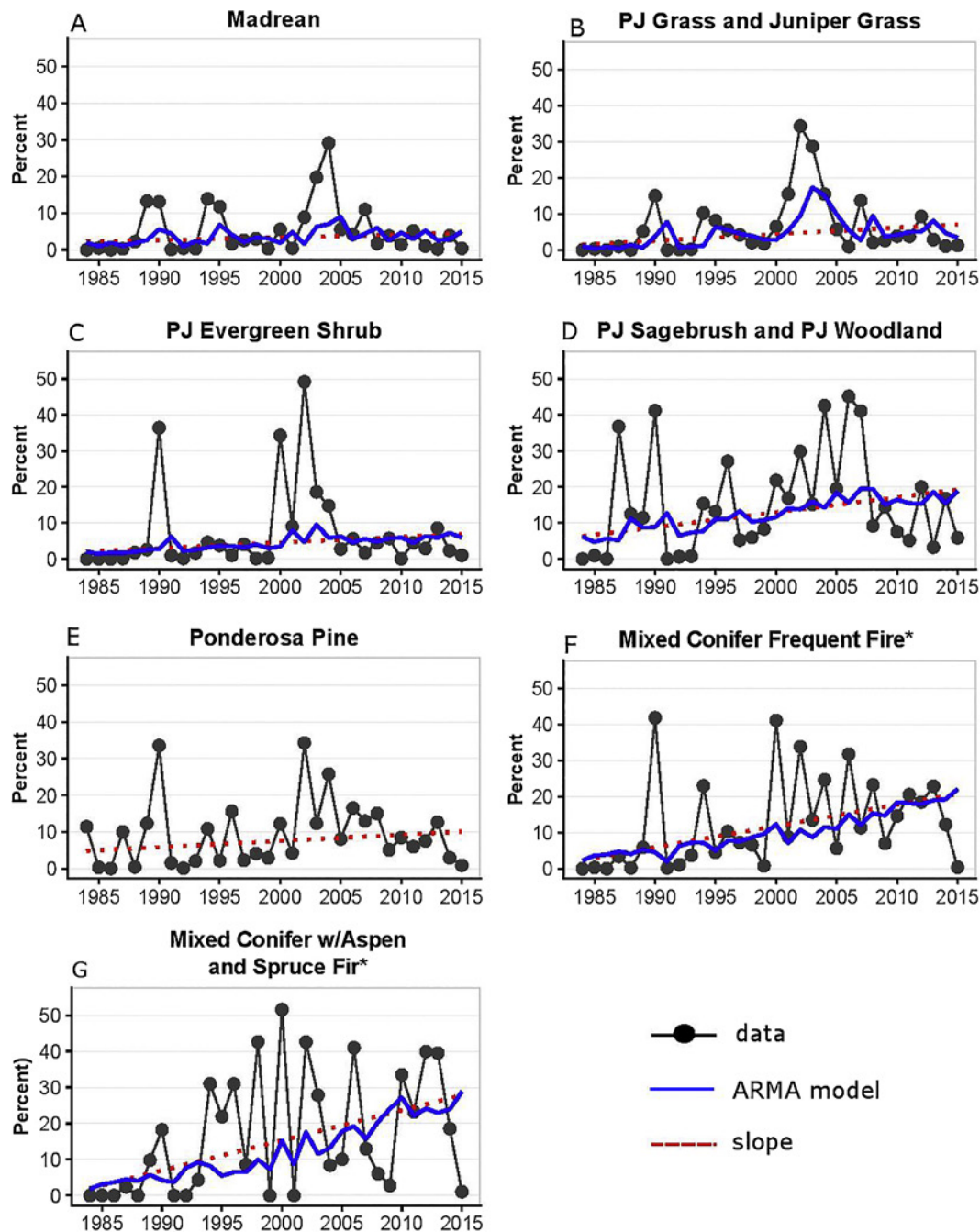


Fig. 6. Results in temporal trend analysis in percent of high severity fire for each ERU Fire Regime Type from 1984 to 2015. Percent high severity fire showed significant increases in two out of the seven ERU Fire Regime Types ($P = 0.001$) indicated by * on graphs, where Madrean, PJ Grass & Juniper Grass, PJ Sagebrush & PJ woodland, PJ Evergreen Shrub, and Ponderosa Pine did not show an increase ($P \geq 0.161$).

fire are uncharacteristic for all ecosystems in the Southwest. In drier forest types such as xeric mixed conifer and ponderosa pine, increased tree densities, fuel loadings, and canopy cover have increased the risk for severe fire as evident in this study (Covington and Moore, 1994; White and Vankjat, 1993). These ecosystems are characterized by low to mixed severity fire regime and thus, increasing trends in high-severity fire may be outside of their natural range of variability (Heinlein et al., 2005). On the other hand, current fire activity may not be abnormal in other ecosystems such as pinyon-juniper, mesic mixed conifer, and spruce-fir due to their longer fire return intervals or incomplete evidence of changing forest structure or fire patterns. However, if the trends observed in this study continue to increase, large-scale changes to the Southwest landscape may be imminent.

Determining whether certain ecosystems are experiencing an alteration in their fire regimes and whether increasing trends in high-severity fire are uncharacteristic will require a thorough examination of historical fire regimes, longer-term remote sensing studies, and how climate and human pressures drive these trends (O'Conner et al., 2014). Further, increasing trends can have a compounding effect on return severity (Holden et al., 2010), forest resilience, and forest recovery (Savage and Mast, 2005) so it will be vital to examine its cumulative effect across the landscape in future studies. It is also important to note that if the trends examined in this study are not uncharacteristic for specific ecosystems, other fire regime attributes such as the spatial patterns (i.e. patch sizes) of high-severity might be. These questions are beyond the scope of this paper and such future research will be

necessary to understand subsequent implications and consequences within each ecosystem.

Documenting trends in burn severity is the first step in understanding contemporary fire regimes, in identifying ecosystems that are vulnerable to uncharacteristic high-severity fire, and managing these ecosystems effectively. Further, quantifying current trends in burn severity and comparing past patterns can serve as a baseline for understanding ecosystem change, identifying where ecosystems have changed the most, and can help us predict future changes in fire regimes. As we progress in our understanding of fire regime-ecosystem interactions, we may need to redefine our land managing strategies in order to meet the challenge of altered fire regimes and altered ecosystem dynamics.

Acknowledgements

Funding provided by Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture. We thank Eric Gdula for providing access to the Grand Canyon National Park's Composite Burn Index dataset. We wish to thank Stephen Howard and Josh Picotte for additional burn severity data needed for this project and their helpful advice as well as Stephanie Mueller for her help in processing Landsat scenes. We also wish to thank Pete Fulé, Doug Cram, and Zander Evans for helpful comments on the manuscript.

Appendix A. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2018.11.039>.

References

- Archibald, S., Scholes, R.J., Roy, D.P., Roberts, G., Boschetti, L., 2010. Southern African fire regimes as revealed by remote sensing. *Int. J. Wildl. Fire* 19, 861–878. <https://doi.org/10.1017/WF10008>.
- Barrett, S., Havlina, D., Jones, J., Hann, W., Frame, C., Hamilton, D., Schon, K., Demeo, T., Hutter, L., Menakis, J., 2010. Interagency fire regime condition class guidebook. Version 3.0. [Homepage of the Interagency Fire Regime Condition Class website] URL: www.frcc.gov.
- Box, G.E.P., Jenkins, G.M., 1976. *Time Series Analysis: Forecasting and Control*, revised ed. Holden-Day, San Francisco, CA.
- Cansler, C.A., McKenzie, D., 2014. Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. *Ecol. Appl.* 24, 1037–1056. <https://doi.org/10.1890/13-1077.1>.
- Chen, J., Zhu, X., Vogelmann, J.E., Gao, F., Jin, S., 2011. A simple and effective method for filling gaps in Landsat ETM+ SLC-off images. *Remote Sens. Environ.* 115, 1053–1064. <https://doi.org/10.1016/j.rse.2010.12.010>.
- Covington, W.W., Moore, M.M., 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. *J. Forest.* 92, 213–220.
- Crimmins, M.A., 2011. Interannual to decadal changes in extreme fire weather event frequencies across the southwestern United States. *Int. J. Climatol.* 31, 1573–1583. <https://doi.org/10.1002/joc.2184>.
- Dennison, P.E., Brewer, S.C., Arnold, J.D., Moritz, M.A., 2014. *Geophys. Res. Lett.* 41, 2928–2933. <https://doi.org/10.1002/2014GL061184>.
- Dillon, G.K., Holden, Z.A., Morgan, P., Crimmins, M.A., Heyerdahl, E.K., Luce, C.H., 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. *Ecosphere* 2, 130. <https://doi.org/10.1890/ES11-00271.1>.
- Eidenshink, J., Schwind, B., Brewer, K., Zhu, Z., Quayle, B., Howard, S., 2007. A project for monitoring trends in burn severity. *Fire Ecol.* 3, 3–21. <https://doi.org/10.4996/fireecology.0301003>.
- Fire Executive Council, 2009. *Guidance for Implementation of Federal Wildland Fire Management Policy*. US Department of Agriculture and US Department of the Interior, Washington, DC, USA.
- Fulé, P.Z., Covington, W.W., Moore, M.M., 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. *Ecol. Appl.* 7, 895–908. [https://doi.org/10.1890/1051-0761\(1997\)007\[0895:DRCFEM\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1997)007[0895:DRCFEM]2.0.CO;2).
- Fulé, P.Z., Korb, J.E., Wu, R., 2009. Changes in forest structure of a mixed conifer forest, southwestern Colorado, USA. *For. Ecol. Manage.* 258, 1200–1210. <https://doi.org/10.1016/j.foreco.2009.06.015>.
- Gdula, E., Brannfors, L., 2014. *Grand Canyon National Park Composite Burn Index Dataset*. Department of the Interior, National Park Service, Grand Canyon National Park.
- Grissino-Mayer, H.D., Swetnam, T.W., 2000. Century scale climate forcing of fire regimes in the American Southwest. *The Holocene* 10, 213–220. <https://doi.org/10.1191/095968300668451235>.
- Hanson, C.T., Odion, D.C., 2014. Is fire severity increasing in the Sierra Nevada, California, USA? *Int. J. Wildl. Fire* 23, 1–8.
- Heinlein, T.A., Moore, M.M., Fulé, P.Z., Covington, W.W., 2005. Fire history and stand structure of two ponderosa pine-mixed conifer sites: San Francisco Peaks, Arizona, USA. *Int. J. Wildl. Fire* 14, 307–320. <https://doi.org/10.1071/WF04060>.
- Hoerling, M.P., Dettinger, M., Wolter, K., Lukas, J., Eischeid, J., Nemani, R., Liebmann, B., Kunkel, K.E., Kumar, A., 2013. Present weather and climate: evolving conditions. In: Garfin, G., Jardine, A., Merideth, R., Black, M., LeRoy, S. (Eds.), *Assessment of Climate Change in the Southwest United States*. NCA Regional Input Reports. Island Press, Washington, DC.
- Holden, Z.A., Morgan, P., Hudak, A.T., 2010. Burn severity of areas reburned by wildfires in the Gila National Forest, New Mexico, USA. *Fire Ecol.* 6, 77–85. <https://doi.org/10.4996/fireecology.0603077>.
- Jolly, W.M., Cochrane, M.A., Freeborn, P.H., Holden, Z.A., Brown, T.J., Williamson, G.J., Bowman, D.M.J.S., 2015. Climate-induced variations in global wildfire danger from 1979 to 2013. *Nat. Commun.* 6, 1–11. <https://doi.org/10.1038/ncomms8537>.
- Kasischke, E.S., Turetsky, M.R., 2006. Recent changes in the fire regime across the North American boreal region - spatial and temporal patterns of burning across Canada and Alaska. *Geophys. Res. Lett.* 33, 1–5. <https://doi.org/10.1029/2006GL025677>.
- Key, C.H., 2006. Ecological and sampling constraints on defining landscape fire severity. *Fire Ecol.* 2, 34–59. <https://doi.org/10.4996/fireecology.0202034>.
- Key, C.H., Benson, N.C., 2006. Landscape assessment (LA): sampling and analysis methods. USDA For. Serv. Rocky Mt. Res. Station. Gen. Tech. Rep. RMS-GTR-164-CD (Fort Collins, CO), pp. 1–55.
- Lee, D.E., Bond, M.L., Borchert, M.I., Tanner, R., 2013. Influence of fire and salvage logging on site occupancy of spotted owls in the San Bernardino and San Jacinto Mountains of Southern California. *J. Wildl. Manage.* 77, 1327–1341. <https://doi.org/10.1002/jwmg.581>.
- Metcalfe, A.V., Cowpertwait, P.S.P., 2009. *Introductory Time Series with R*. Springer Science & Business Media.
- Miller, J.D., Safford, H., 2012. Trends in wildfire severity: 1984 to 2010 in the Sierra Nevada, Modoc Plateau, and Southern Cascades, California, USA. *Fire Ecol.* 8, 41–57. <https://doi.org/10.4996/fireecology.0803041>.
- Miller, J.D., Safford, H.D., Crimmins, M., Thode, A.E., 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. *Ecosystems* 12, 16–32. <https://doi.org/10.1007/s10021-008-9201-9>.
- Miller, J.D., Skinner, C.N., Safford, H.D., Knapp, E.E., Ramirez, C.M., 2012. Trends and causes of severity, size, and number of fires in northwestern California, USA. *Ecol. Appl.* 22, 184–203.
- Miller, J.D., Thode, A.E., 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). *Remote Sens. Environ.* 109, 66–80. <https://doi.org/10.1016/j.rse.2006.12.006>.
- Moore, M.M., Huffman, D.W., Fulé, P.Z., Covington, W.W., Crouse, J.E., 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. *For. Sci.* 50, 162–176.
- MTBS [Monitoring Trends in Burn Severity], 2013. *Monitoring Trends in Burn Severity (MTBS)*. US Department of Agriculture, Geological Survey and US Department of Agriculture, Forest Service. < <http://www.mtbs.gov> > (accessed 02.12.16).
- Neary, G., Klopatek, C., DeBano, L., Ffolliott, P., 1999. Fire effects on belowground sustainability: a review and synthesis. *For. Ecol. Manage.* 122, 51–71.
- NOAA [National Oceanic and Atmospheric Administration], 2018. Climate at a Glance < <http://www.ncdc.noaa.gov/cag/> > (accessed 04.20.16).
- Notaro, M., Liu, Z., Gallimore, R.G., Williams, J.W., Gutzler, D.S., Collins, S., 2010. Complex seasonal cycle of ecohydrology in the Southwest United States. *J. Geophys. Res. Biogeosci.* 115, G04034. <https://doi.org/10.1029/2010JG001382>.
- O'Conner, C.D., Falk, D.A., Lynch, A.M., Swetnam, T.W., 2014. Fire severity, size, and climate attributes diverge from historical precedent along an ecological gradient in the Pinaleno Mountains, Arizona, USA. *For. Ecol. Manage.* 329, 264–278. <https://doi.org/10.1016/j.foreco.2014.06.032>.
- Picotte, J.J., Peterson, B., Meier, G., Howard, S.M., 2016. 1984–2010 trends in fire burn severity and area for the conterminous US. *Int. J. Wildl. Fire* 25, 413–420.
- Reilly, M.J., Dunn, C.J., Meigs, G.W., Spies, T.A., Kennedy, R.E., Bailey, J.D., Briggs, K., 2017. Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). *Ecosphere* 8. <https://doi.org/10.1002/ecs2.1695>.
- Riano, D., Moreno Ruiz, J.A., Isidoro, D., Ustin, S.L., 2007. Global spatial patterns and temporal trends of burned area between 1981 and 2000 using NOAA-NASA Pathfinder. *Glob. Chang. Biol.* 13, 40–50. <https://doi.org/10.1111/j.1365-2486.2006.01268.x>.
- Rivera-Huerta, H., Safford, H.D., Miller, J.D., 2016. Patterns and trends in burned area and fire severity from 1984 to 2010 in the Sierra de San Pedro Mártir, Baja California, Mexico. *Fire Ecol.* 12, 52–72. <https://doi.org/10.4996/fireecology.1201052>.
- Rollins, M.G., Frame, C.K., 2006. The LANDFIRE Prototype Project: Nationally consistent and locally relevant geospatial data for wildland fire management. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. General Technical Report RMRS-GTR-175. <https://doi.org/10.1016/j.foreco.2009.07.051>.
- Safford, H.D., Miller, J.D., Collins, B.M., 2015. Differences in land ownership, fire management objectives and source data matter: a reply to Hanson and Odion (2014). *Int. J. Wildl. Fire* 24, 286–293. <https://doi.org/10.1071/WF14013>.
- Savage, M., Mast, J.N., 2005. How resilient are southwestern ponderosa pine forests after crown fires? *Can. J. For. Res.* 35, 967–977. <https://doi.org/10.1139/x05-028>.
- Sheppard, P.R., Comrie, A.C., Packin, G.D., Angersbach, K., Hughes, M.K., 2002. The climate of the US Southwest. *Clim. Res.* 21, 219–238. <https://doi.org/10.3354/cr021219>.
- Shumway, R.H., Stoffer, D.S., 2010. *Time Series Analysis and Its Applications: With R*

- Examples, third ed. Springer Science & Business Media.
- Sokal, R.R., Rohlf, J.F., 1981. *Biometry: The Principles and Practice of Statistics in Biological Research*, second ed. W. H Freeman and Company, San Francisco, pp. 859.
- Stephens, S.L., 2005. Forest fire causes and extent on United States Forest Service lands. *Int. J. Wildl. Fire* 14, 213–222. <https://doi.org/10.1071/WF04006>.
- Turner, M.G.M., Hargrove, W.W., Gardner, R.H., Romme, W.H., 1994. Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. *J. Veg. Sci.* 5, 731–742. <https://doi.org/10.2307/3235886>.
- Wahlberg, M., Triepke, F., J., Robbie, W., Stringer, S.H., Vandendriesche, D., Muldavin, E., Malusa, J., 2014. *Ecological Response Units of the southwestern United States*. USDA Forest Service Forestry Report FR-R3-XX-XX. Southwestern Region, Regional Office, Albuquerque, NM, 201pp.
- Westerling, A.L., 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of the spring. *Philos. Trans. R. Soc. B* 371, 20150178. <https://doi.org/10.1098/rstb.2015.0178>.
- Williams, A.P., Allen, C.D., Macalady, A.K., Griffen, D., Woodhouse, C.A., Meko, D.M., Swetnam, T.W., Rauscher, S.A., Seager, R., Grissino-Mayer, H.D., Dean, J.S., Cook, E.R., Gangodagamage, C., Cai, M., McDowell, N.G., 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. *Nat. Clim. Change* 3, 292–297.
- White, M.A., Vankjatt, J.L., 1993. Middle and high elevation coniferous forest communities of the North Rim region of Grand Canyon National Park, Arizona, USA. *Vegetatio* 109, 161–174.
- Yocom-Kent, L.L., Fulé, P.Z., Bunn, W.A., Gdula, E.G., 2015. Historical high-severity fire patches in mixed-conifer forests. *Can. J. For. Res.* 45, 1587–1596. <https://doi.org/10.1139/cjfr-2015-0128>.